9.5.3 Excavation

Excavation technologies are described below. Soils contaminated above PRGs would be selectively excavated, to the extent feasible, using field-screening technologies to identify radionuclide activities, and toxic metal concentrations, exceeding PRGs. Instrumentation could include hand-held sodium iodide detectors for Cs-137, which were successfully used for the INEEL OU 10-06 radioactive soil consolidation; and X-ray fluorescence (XRF) spectrometers for toxic metals, also available on the INEEL. Comparing current reported XRF detection limits for Pb, Cu, and Hg (50, 50, and 200 mg/kg, respectively) (Ashe et al. 1991) to the PRGs reported in Table 9-3 indicate that the instrument could be effective for determining the extent of Pb and Cu contamination above the PRGs, but would not be useful for Hg.

9.5.3.1 Backhoes and Dozers. These process options represent standard excavation techniques utilizing conventional equipment. Conventional equipment has been demonstrated to be completely effective for removing contaminated soil to depths of at least 8 m (25 ft) bgs at the INEEL, and potentially to greater depths depending on site conditions. Equipment operators can be shielded in positive pressure cabs as needed to reduce exposures during excavation. Impacts to human health and the environment could likely be minimized to allowable levels through administrative and engineering controls. These process options are therefore considered technically and administratively feasible. Costs are considered to be relatively low.

9.5.3.2 Robotics. This process option represents non-standard excavation techniques using remote-operated equipment. These technologies are not globally demonstrated to be effective and implementable, and would have to be evaluated on a site-specific basis. No OU 4-13 soil contamination has been determined to be classified as remote handled waste (>500 mR/hr at 0.9 m [3 ft] in air); therefore, robotics are likely not required to reduce worker exposures to allowable levels. Costs are considered relatively high. This technology was therefore screened from further consideration on the basis of cost-effectiveness.

9.5.4 Containment

9.5.4.1 ET-Type (Capillary Barrier/Biobarrier) Cover. This technology is estimated to be highly effective in protecting human health and the environment and meeting RAOs for OU 4-13. The capillary barrier/biobarrier cap, shown in Figure 9-1, consists of layers of fine-grained earthen materials overlying coarse-grained media. The large variation in soil moisture tension between the two layers results in infiltrating water being retained in the upper, fine-grained layers by capillary attraction, within the root zone of surficial vegetation, until saturated. Evaporation and plant transpiration can remove essentially all precipitation that falls in arid regions, including the INEEL high desert environment (Anderson et al. 1992), typically preventing development of saturated conditions and preventing drainage through the capillary barrier (Keck et al. 1992). A base course of asphalt or concrete may be used to further limit infiltration. The capillary break would also serve as a biobarrier, inhibiting biointrusion, or alternatively a separate layer can be used for this function.

Several variations of the ETC design are currently undergoing field testing at the INEEL Radioactive Waste Management Complex (RWMC) Subsurface Disposal Area (SDA) (Anderson 1997a 1997b, and Bhatt and Porro 1998), and have been tested successfully at Hanford and Los Alamos (Nyhan et al. 1990). The Hanford Permanent Isolation Surface Barrier (PISB) includes multiple low-permeability base courses, as well as a 1.5 m (5 ft) thick fractured basalt layer that serves as a capillary break, biointrusion and human intrusion barrier. Many variations on the basic engineering technology (ET) design are possible, depending on functional and operational requirements for the site.

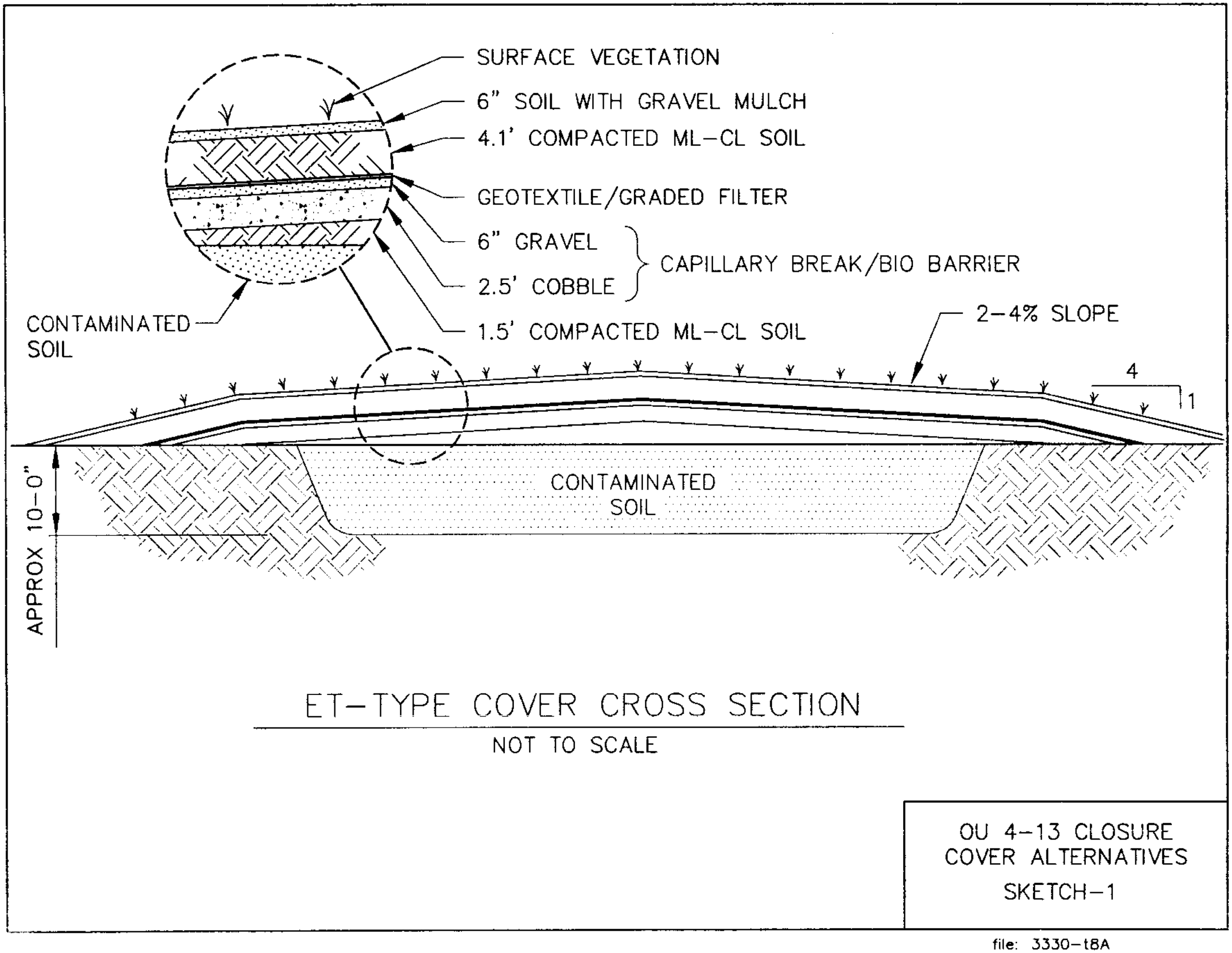


Figure 9-1. Cross-section of the ET-type cover or barrier.

Overlying fine soil must be prevented from entering the coarser underlying media, to maintain the function of both the biobarrier and capillary barrier components. If fine soil fills the coarser media, it can serve as a conduit for both infiltrating water and for plant roots (Keck et al. 1992). Geotextile or a graded filter bed would be placed over the biobarrier to prevent fine soil intrusion.

This cover or barrier was designed to control surface exposures and inhibit biotic intrusion and infiltration for at least 500 to 1,000 years. Impacts to human health and the environment could likely be minimized to allowable levels through administrative and engineering controls. The cover has been constructed at pilot-scale and is therefore considered technically implementable. The relative cost of this cover is moderate. This option is retained for further consideration.

9.5.4.2 Native Soil Cover. This cover type consists of native INEEL soil compacted in lifts and covered with vegetation, gravel, riprap or other media. This design is completely effective in controlling surface exposures, but is not as effective in inhibiting biointrusion as the engineered cover, since maximum rooting depths of native INEEL vegetation including Big Sagebrush can exceed 3 m (10 ft). Sagebrush and potentially other plant species could therefore uptake and mobilize contaminants in the food chain.

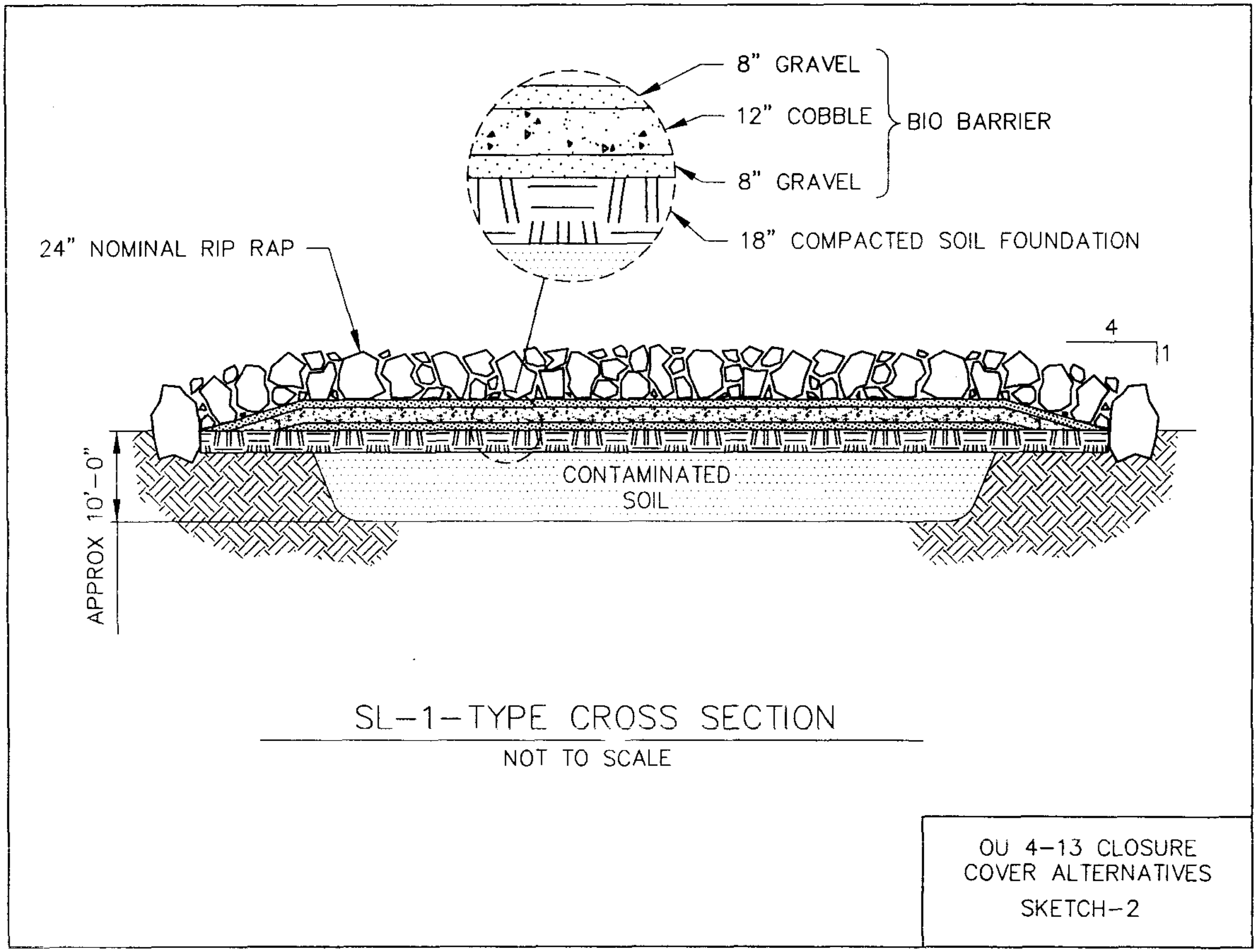
Soil covers are readily implementable and have been previously applied at the INEEL. Impacts to human health and the environment during construction could likely be minimized to allowable levels through administrative and engineering controls. The relative cost of this cover is low to moderate.

This process option also represents backfilling of disposal ponds, consisting of filling contaminated ponds in lifts with clean INEEL native soil, grading, and covering the surface with vegetation or other media. This option is retained for further consideration.

9.5.4.3 SL-1-Type Barrier. The SL-1-type barrier, shown in Figure 9-2, consists of a layer of basalt cobbles approximately 30 cm (12 in.) thick, underlain and overlain by gravel (10 cm [4 in.] thick), with a rock armor surface (0.6 m [2 ft] thick). Overall thickness is approximately 1.2 m (4 ft). This type of barrier was designed to control surface exposures and erosion at uranium mill tailings remedial action (UMTRA) sites for at least 500 years. A biobarrier was added to the SL-1-type design to inhibit biotic intrusion. This barrier is estimated to be effective in reducing human health risks via direct radiation exposure, soil ingestion, and homegrown produce ingestion. Additionally, INEL (1995) determined that this barrier would effectively limit biointrusion, thereby limiting ecological risks.

However, this barrier does not reduce infiltration, does not promote runoff of rainfall and snowmelt, and does not promote lateral drainage of infiltration, which are typical functions of a closure cover. This barrier will likely increase infiltration rates, relative to undisturbed soils, since any rainfall or snowmelt on the barrier rapidly moves through the depth of the very porous rock armor and gravel-cobble layers, beyond the depth of evaporation. Transpiration would not remove water, since no vegetation would be present. This barrier therefore would likely increase risks due to infiltration and leaching of COCs to groundwater, by increasing COC migration through the soil column. Note that GWSCREEN modeling performed for OU 4-13 sites showed no significant groundwater risks resulting from surface infiltration, using a default infiltration rate of 10 cm/yr.

Wind-transported fine soil must be prevented from entering the coarse biobarrier media to maintain the function of the biobarrier. If fine soil fills the gravel and cobble layers, it will serve as a conduit for plant roots (Keck et al. 1992). Geotextiles or a graded filter bed should be placed atop the biobarrier to prevent fine soil intrusion.



file: 3300-t7A

Figure 9-2. Cross-section of the SL-1-type barrier.

This barrier has been built on the INEEL at sites where infiltration and leaching to groundwater is not a concern, and is therefore highly technically and administratively implementable. Impacts to human health and the environment could likely be reduced to allowable levels through administrative and engineering controls. The cost of this cover is low to moderate.

9.5.4.4 RCRA-Type Cover. This type of cover was developed for closing lined RCRA Subtitle C landfills, and consists of three layers including: (1) a low-permeability bottom layer consisting of a geosynthetic flexible membrane overlying a compacted clay layer, (2) a middle drainage layer consisting of sand or geosynthetic drainage net, separated from the overlying layer by a geosynthetic or earthen filter layer, and (3) an upper vegetated layer consisting of soil with grass planted at the surface. The cover is typically built with a 3 to 5% slope at the surface and at layer interfaces, to promote runoff and lateral drainage, and a 4:1 side slopes (EPA 1989). This type of cover is typically 1.8 to 2.4 m (6 to 8 ft) thick.

This design was primarily designed to promote runoff and lateral drainage, and control infiltration, in humid climates. This design is effective in controlling surface exposures and infiltration; however, compacted clay layers have been determined to suffer from desiccation cracking resulting in permeability increases, both during and after construction, particularly in arid regions. Geomembranes may also fail during installation, or due to subsidence, and a design life has not been defined for this type of cover. This design is not highly effective in inhibiting biointrusion, but a biobarrier can be incorporated into the design if required. Constructing the cover is moderately implementable, due to the added complexity of using geosynthetics and compacted clay. Impacts to human health and the environment could likely be minimized to allowable levels through administrative and engineering controls. The cost of this cover is relatively high.

9.5.4.5 Concrete. Concrete is used or planned for use in NRC-regulated low-level waste (LLW) disposal facility elements including disposal vaults, backfill, and closure covers. The primary reason for using concrete in closure covers is to meet NRC requirements for 500 years of post-closure inadvertent intruder protection. While some concrete structures have survived for centuries, concrete is susceptible to damage or attack including:

- Physical damage (including cracking) as a result of subsidence, freeze-thaw action, seismic activity, erosion and abrasion, etc.

- Chemical attack by sulfate, chloride, alkali-aggregate reaction, leaching, acid attack, carbonation, etc. (use of additives may reduce or eliminate some forms of chemical attack)

- Other degradation processes including biodegradation and irradiation (Walton et al. 1990 and 1991).

Concrete with a low water:cement ratio can have a hydraulic conductivity of less than 1E-12 cm/second. However, the actual permeability of weathered concrete structures is dominated by cracks, and therefore the permeability will increase over time as weathering occurs (Walton et al. 1991). Sulfur-polymer cements are potentially more resistant to chemical attack by acids and salts, and are also mechanically stronger. Risks of cracking due to landfill subsidence have limited use of concrete for landfill covers.

Concrete would be effective for eliminating direct radiation exposures, soil ingestion, and inhibiting biointrusion. The thickness of the cover could be scaled to the area of the site, shielding requirements, etc. Implementability is higher for smaller than for larger sites. Costs are relatively high.

9.5.5 Disposal

9.5.5.1 RWMC. Disposal of radionuclide-contaminated soil and debris at the RWMC is completely effective in protecting human health and the environment and in meeting RAOs. RCRA-regulated hazardous waste cannot be disposed of at the RWMC; however, LLW resulting from treating mixed waste is allowed for disposal, if the waste is not listed, does not exhibit characteristic hazards and meets all LDR treatment standards. Disposal requirements for contact handled LLW are stated in the INEEL reusable property, recyclable materials, and waste acceptance criteria (RRWAC) (DOE 1998). Characterization requirements include quantitation of specific radionuclides. Soils may be added to fill voids in waste containers, if a plan for this type of disposal is submitted and approved. Bulk disposal of soil is not currently allowed, and the RRWAC contains a list of allowed containers.

This option has been used for prior INEEL CERCLA actions and is therefore readily considered technically and administratively feasible. Impacts to human health and the environment could likely be minimized to allowable levels through administrative and engineering controls. Currently, RWMC operations discourage disposal of low-level radioactively contaminated soils. However, there is no stated INEEL policy preventing the practice, so the option is retained. An estimated 55,813 m^3 (73,000 yd^3) of disposal capacity remain at the RWMC. Costs are relatively high, if stated disposal costs are applied.

9.5.5.2 INEEL CERCLA Disposal Facility. An INEEL soil repository (the ICDF), projected to be located south of the INTEC, would begin accepting INEEL CERCLA and environmental restoration (ER) soil and debris contaminated with radionuclides and/or other COCs in 2001, and RCRA-hazardous and/or mixed wastes in 2002. The preconceptual design of this facility includes sufficient disposal capacity to accept all such remediation wastes from all INEEL WAGs, including those from WAG 4.

The effectiveness and implementability of this option are uncertain, due to the conceptual status of the project, but the option is retained for further consideration pending a final decision. Projected disposal costs for this facility are much lower than those for the RWMC or offsite low-level radionuclide-contaminated soil and debris landfills.

9.5.5.3 INEEL Landfill Complex. This option could include disposal in the currently operational CFA landfill, or backfilling an existing excavation or disposal pond (e.g., the CFA-04 pond). Soils disposed of in either type of unit would consist primarily of material contaminated with toxic metals at levels above ecological PRGs but which pass the TCLP, and which contain radionuclide concentrations below action levels. The existing CFA landfill could also accept some treated RCRA characteristic wastes (e.g., D004-D011 metal wastes treated to 40 CFR 268.40 standards). The existing CFA landfill is projected to continue to operate at least 10-15 years in the future[b]. Soils disposed of there must meet sections of the RRWAC (DOE 1998) applicable to industrial wastes, as well as state and federal regulations.

Characterization requirements would likely be minimal for this alternative, and would likely be met by characterization performed during excavation, as well as process knowledge. The CFA landfill accepts bulk shipments of industrial wastes; therefore, no containerization would be required.

The CFA-04 pond could potentially be backfilled with contaminated soil and capped. This option was previously used for INEEL OU 10-06 radionuclide-contaminated soils, which were consolidated in the Test Reactor Area (TRA) Warm Waste Pond (WWP). The estimated volume to fill the CFA-04 pond

b. M. Wraught, personal communication.

is approximately 3.5E+04 m^3 (4.6E+04 yd^3), which would be adequate to contain all contamination at OU 4-13, except for the CFA-08 drainfield soils, which are estimated at 5.7E+04 m^3 (7.4E+04 yd^3).

The effectiveness of this option, if combined with an effective cover on the disposal unit, is considered high. This option is considered technically and administratively implementable. Costs are estimated as low.

9.5.5.4 Offsite Mixed Low-Level Waste Landfill. This option is considered highly effective in protecting human health and the environment and in meeting RAOs. A facility of this type exists approximately 482 km (300 mi) south of the INEEL. The facility is supported by a rail spur, with a railcar rollover, allowing for bulk shipment from the INEEL directly by rail. This facility is permitted to accept low-level radioactive soils, and can treat and dispose of some mixed waste soils.

Prior to disposal at this facility, generators must submit a pre-shipment sample profile record form, then submit a minimum of five, 0.9 kg (2 lb) diverse, representative samples per waste stream. Waste acceptance criteria for this facility include analyses for gamma spectroscopy (natural and man-made isotopes), uranium and thorium isotopic analyses, full TCLP analysis (EPA Method 1311), total metals and total organic, hydrogen sulfide and hydrogen cyanide reactivity, pH, paint filter test for free liquids, additional analyses for individual waste code LDRs, a listed waste evaluation, and a Proctor test (ASTM D-698).

Maximum and average radionuclide activities in OU 4-13 remediation waste were compared to maximum activity allowed under the representative off-site disposal facility's operating permit. All average measured activities are less than allowed levels.

Potential RCRA waste codes for OU 4-13 remediation waste could include D008 and D009. The representative mixed low-level waste (MLLW) treatment, storage, and disposal facility (TSDF) is permitted for treatment and disposal of both of these waste codes (D008 low mercury subcategory only).

Impacts to human health and the environment could likely be minimized to allowable levels through administrative and engineering controls. The INEEL soils have previously been shipped to this facility for disposal; therefore, this option is considered technically and administratively implementable. Costs for this option are estimated as moderate-high, relative to other disposal options.

9.5.5.5 RCRA TSDF. This option is considered highly effective in protecting human health and the environment and in meeting RAOs for soils contaminated with RCRA-characteristic wastes. The RCRA Subtitle C facilities approximately 482 km (300 mi) from the INEEL at Clive, Utah; and another approximately 1,287 km (800 mi) away at Arlington, Oregon, have previously been used for treatment and disposal of RCRA-contaminated soils.

Prior to disposal at this facility, generators must submit a pre-shipment sample profile record form, then submit representative samples for each waste stream. Waste acceptance criteria include full TCLP analysis (EPA Method 1311), total metal and total organic, hydrogen sulfide and hydrogen cyanide reactivity, pH, paint filter test for free liquids, a listed waste evaluation, and additional analyses for individual waste code LDRs.

Impacts to human health and the environment could likely be reduced to allowable levels through administrative and engineering controls. This process option is therefore considered technically and administratively implementable. Costs for this option are estimated as high, relative to other options.

9.5.5.6 Nevada Test Site. The Nevada Test Site (NTS) is permitted to receive defense low-level and mixed radioactive wastes. The NTS is approximately 970 km (600 mi) southwest of the INEEL and is not serviced directly by rail. Waste transport from the INEEL would be directly by truck, or with transfer from rail cars to trucks near Las Vegas, NV. Currently, the INEEL is not an approved generator for disposal at NTS, and this issue remains unresolved. This option is screened from further consideration as not currently administratively implementable.

9.5.6 Treatment Ex Situ

Ex situ treatment options can be performed on excavated contaminated media, and can be performed onsite or offsite. Several treatment options for INEEL soils and sediments, including physical, chemical, and thermal technologies, have been investigated at bench- and in some cases pilot-scale. The objectives of treatment at CERCLA sites are primarily to reduce the toxicity, mobility, and volume of contaminated media. Toxicity of radionuclides is only reduced by natural radioactive decay, toxicity of toxic metals in most cases cannot be reduced, and organics can be destroyed and toxicity eliminated. Mobility of COCs could be reduced through immobilization in a stable matrix. Volume of contaminated media may also be reduced by ex situ treatment.

Effectiveness of many soil treatment options is very site-specific and depends on contaminants present, soil textural classification, mineralogy, chemistry, and many other factors. Evaluations of effectiveness of treatment options in this FS include technologies evaluated and demonstrated for soils and COCs found at the INEEL, and at the CFA.

Construction requirements may include excavating and transporting contaminated media, constructing above ground process equipment, and other activities. Ex situ treatment options potentially applicable to OU 4-13 sites of concern are discussed below.

9.5.6.1 Physical Separation Using Screening. This technology takes advantage of the typical tendency of radionuclides and heavy metals to be distributed more into soil fines (silts and clays) than into coarse components (coarse sands, gravels, and cobbles). This is often the most effective separation step in a soil washing process. Excavated, contaminated soils can be passed through progressively finer screen sizes, using grizzly shakers or other standard process equipment, to separate fine-grained from coarse-grained fractions. This technology may be used alone or in combination with other treatment technologies to reduce the volume of contaminated soils for disposal.

This technology was tested for Cs-137 separation in treatability studies using TRA WWP sediments and soils (DOE 1995b). Tests determined that this process is effective at separating fine-grained from coarse-grained fractions. However, the effectiveness of screening in reducing the volume of contaminated soils is likely to be limited because Cs-137 in WWP sediments and soils is apparently not sufficiently concentrated in the fine-grained fraction to result in separation of a soil fraction that could be returned to the site (i.e., for which Cs-137 concentrations were less than 16.7 pCi/g, the remedial goal [RG] at the time). About 30% of the total cesium present was in the greater than 8 mesh material (gravel and cobbles), which represented at least 60% by weight of the WWP sample sediments.

This technology has not been tested for separating OU 4-13 toxic metal COCs including Pb, Hg, and Cu from INEEL soils. If metals fractionated significantly into clays and silts, significant volume reduction could be achieved. Effectiveness could only be determined in treatability studies.

Impacts to human health and the environment during operations could likely be reduced to allowable levels through administrative and engineering controls. This option is technically implementable using standard process equipment. Costs are relatively low. This technology is retained for further consideration.

9.5.6.2 Physical Separation Using Flotation. Flotation separates fine-grained from coarse-grained soils based on their different settling velocities in a water clarifier. This technology is often used in a soil washing process. The soils would be added to a conical tank filled with water, and air introduced into the tank through diffusers or impellers. The bubbles attach to the particulates and the buoyant forces on the combined particle and air bubbles are sufficient to cause fine-grained particles to rise to the surface where they can be recovered by skimmers. Coarse-grained materials are removed from the bottom of the tank.

This technology was tested in treatability studies using TRA WWP sediments and soils. Tests determined that this process is effective at separating fine-grained from coarse-grained fractions. However, the effectiveness of flotation in reducing the volume of contaminated soils was limited, again because Cs-137 distribution in WWP sediments and soils apparently is not sufficiently concentrated in the fine-grained fraction to result in separation of a soil fraction that could be returned to the site (i.e., for which Cs-137 concentrations were less than 16.7 pCi/g, the RG at the time). This technology also produces a secondary liquid waste stream; however, the water may be reusable after treatment.

This technology has not been tested for separating OU 4-13 toxic metal COCs including Pb, Hg, and Cu from INEEL soils. If metals fractionated significantly into clays and silts, significant volume reduction could be achieved. Effectiveness could only be determined in treatability studies.

Impacts to human health and the environment during operations could likely be reduced to allowable levels through administrative and engineering controls. This option is considered moderately technically implementable, due to increased process complexity and requirements for secondary waste handling. Costs are relatively moderate. This technology is retained for further consideration.

9.5.6.3 Physical Separation Using Attrition Scrubbing. Attrition scrubbing consists of mechanical agitation of soil and water mixtures in a mixing tank, to remove contaminants bound to external particle surfaces. This technology was not determined to be effective for cesium removal from WWP sediments and soils (DOE 1995b), because only 18% of the cesium was determined to be associated with phases in and on the sediment particle coatings. The remaining 82% was determined to be associated with the particle internal mineral lattice structure and could be removed only by dissolution of the particle. However, this technology, combined with screening, was estimated to be potentially effective for soils with initial activities within 10 times the RG at the time (i.e., 167 pCi/g). Further treatability studies on representative samples from CFA contaminated soil sites would be required to determine the effectiveness of this technology, alone or in combination with others, to reduce the volume of contaminated soils.

This technology has not been tested for separating OU 4-13 toxic metal COCs including Pb, Hg, and Cu from INEEL soils. Effectiveness could only be determined in treatability studies.

Impacts to human health and the environment during operations could likely be reduced to allowable levels through administrative and engineering controls. This technology also produces a secondary liquid waste stream. Costs are estimated as relatively moderate. The effectiveness of attrition scrubbing for reducing the volume of contaminated materials at OU 4-13 sites is low to uncertain. This option is retained for further consideration.

9.5.6.4 Soil Washing. This technology may include various combinations of physical treatment processes, discussed previously, and chemical processes discussed in this section. This option would consist of physically and chemically extracting contaminants from excavated soils and debris to produce clean soils and concentrated residuals. Clean soils could be returned to the site of concern and concentrated stabilized residuals would likely be landfilled. Extractants could include water, acids, surfactants, brines, carbonates, or other compounds.

Soil washing using water and concentrated nitric acid, in combination with screening, attrition scrubbing and flotation, has previously been tested at bench-scale on TRA WWP sediments with poor results. Although cesium removal efficiency for WWP sediments for the greater than 8 mesh fraction (gravels and cobbles) exceeded 90%, cesium activity in the treated solids still exceeded the 690 pCi/g test treatment goal (INEL 1991a, WINCO 1994, and DOE 1995b). Based on these results, little or no volume reduction of Cs-137 contaminated materials would be achieved using this combination of methods for CFA soils.

Note that the TRA treatability studies were performed on WWP samples. Other native soils and disposal pond sediments at the INEEL may differ in composition and mineralogy.

No soil washing treatability studies have been performed to date using toxic metal-contaminated INEEL soils. The contaminants Pb, Cu, and Hg, as well as radionuclides including U-238 and U-235, have successfully been removed from soils at other sites, including Hanford, using a combination of screening, flotation, and extraction. Much of the volume reduction occurred at the screening step (EPA 1995). Treatability studies would be required to determine effectiveness for Pb, Cu, and Hg in OU 4-13 soils.

Toxicity of the radionuclides and/or toxic metals would not be reduced. This technology would produce large volume secondary waste streams requiring treatment, however in some cases the extractant may be reused, reducing secondary waste volume. The effectiveness of soil washing for reducing risks to human health and the environment and meeting RAOs at OU 4-13 is uncertain. Based on unsuccessful soil washing tests performed on Cs-137-contaminated INEEL soils, and on no prior soil washing tests for toxic metals in INEEL soils, soil washing cannot be determined to significantly improve protection of human health and the environment, or to reduce volumes of contaminated materials, at OU 4-13 sites. Additionally, the relatively small size of the OU 4-13 toxic-metal contaminated sites (CFA-04 and -10) would limit the cost effectiveness of this technology, since significant costs would be incurred for treatability studies and capital equipment. Impacts to human health and the environment could likely be minimized to allowable levels through administrative and engineering controls.

The implementability of this option is considered low to moderate, based on the requirements for soil- and contaminant-specific treatability studies, the potential complexity of the process, and the potentially large volumes of secondary wastes produced. Costs are considered moderate relative to other ex situ treatment technologies. This option is screened from further consideration on the basis of low cost effectiveness.

9.5.6.5 Physical Separation Using a Segmented Gate System. This technology would apply only to CFA-08, where Cs-137 is present above human health PRGs. The system combines a feed hopper, conveyer belt, real-time gamma spectroscopy and a series of movable gates to separate soils moving on the belt on the basis of activity. Other unit processes including crushing, screening, and sizing may be required to produce relatively uniformly sized feeds. This technology is currently under consideration for the INEEL Pit 9 removal and treatment project to reduce the volume of excavated material requiring treatment prior to final disposal. Materials above and below allowable activities are diverted to different outlets. Soils with radionuclide activities below allowable levels could be returned to the excavation, while soils with radionuclide activities above allowable levels could be treated further or directly disposed of at an appropriate landfill.

The effectiveness of this technology for OU 4-13 soils and sediments is uncertain and would require field demonstration. This technology has been successfully demonstrated to reduce volumes of radionuclide-contaminated soils at several sites and a field demonstration at the INEEL for separating contaminated soils at the 23 pCi/g Cs-137 PRG is planned for 1999.

This technology does not produce significant secondary waste streams and mainly utilizes conventional material-handling equipment. Gamma radiation detectors may be either germanium or sodium iodide. The gamma monitoring-conveyer and gate system may be combined with other technologies in a treatment train, for example vitrification, to stabilize the soils and sediments containing the highest activities. This option is most applicable to sites where soils have not been disturbed after contamination (i.e., where contaminants have not been homogenized in the soil). These types of sites may include those with wind- and water-deposited contamination. This technology is likely less effective for sites where contaminated soils have been previously consolidated.

Previous uses at Johnson Atoll and at Savannah River claimed high volume reductions; however, effectiveness depends on soil type. Contaminants at Johnson Atoll included particulate radionuclides dispersed in coral carbonate soil. The contaminated soils at Savannah River were encountered during excavations for new construction and ongoing operations. The Savannah River system initially physically screened material greater than approximately 5 cm (2 in.) for separate counting. Some of the reported volume reduction was apparently due to size separation prior to processing through the scanning gate; however, the separation efficiency of sizing alone was not reported. The system operational detection limit was determined to be 2.4 pCi/g. The release criterion was defined as 4 pCi/g; therefore, the system met this requirement. Approximately 1,200 m^3 (1,570 yd^3) of soil were processed, with a reported volume reduction of 99.3%. Average processing rates were approximately 25 m^3/hour (33 yd^3/hour). Only one sorter system and four personnel were used. Reported mobilization/demobilization costs are $25,000 per sorter. Reported processing costs are $35 to $60 per yd^3 (TMA/Eberline 1995)

Impacts to human health and the environment during operations could likely be reduced to allowable levels through administrative and engineering controls. This technology is considered moderately implementable. Costs are estimated as relatively moderate. This technology is retained for further consideration pending an INEEL field demonstration for Cs-137-contaminated soil.

9.5.6.6 Chemical Stabilization. This option would consist of adding chemical amendments such as Portland cement, polymers, pozzolons, calcium or sodium silicates, or other amendments to excavated soils to produce a stable wasteform. Immobilization of contaminants may occur by formation of an insoluble chemical species, or by microencapsulation in the matrix of the wasteform. This option alone would not significantly reduce risks due to direct radiation exposure that are relative to unstabilized soil. Toxicity of the radionuclides and/or toxic metals would not be reduced; however, availability of COCs and exposure risks via soil ingestion and plant uptake would be reduced. Disposal of the wasteform in a low-level radioactive or mixed waste soil and debris landfill would likely be required. Mobility via leaching and infiltration to groundwater would be reduced. Volume of contaminated materials would increase by up to 200%.

This technology might be used after radionuclide separation using a segmented gate or other system, to produce a stable wasteform for disposal of relatively high concentration solids. However, it is unlikely if any soil fractions from separation processes at CFA would be of high enough activity to require stabilization prior to disposal.

Stabilization in Portland cement, potentially with amendments including calcium silicate, fly ash, or others, would likely meet RCRA LDR treatment standards for OU 4-13 toxic metals including Hg and Pb, based on previous stabilization studies at the INEEL (Gering and Schwendiman 1996) and elsewhere. However, some species of Pb and Hg are difficult to stabilize, and Portland cement stabilization may not provide long-term contaminant immobilization (Mattus and Gilliam 1994). In general, the duration of contaminant isolation for chemical stabilization is undefined, but is significantly less than for vitrification.

Treatability studies would be required to determine effectiveness. Overall, this technology may offer little improvement in effectiveness over excavation and disposal in a secure landfill without treatment, but it is more administratively implementable for RCRA characteristic wastes.

Impacts to human health and the environment could likely be minimized to allowable levels through administrative and engineering controls. The technical implementability of this option is considered moderate. Extensive handling and mixing of the soils would be required to produce a homogeneous wasteform. However, standard construction and soil handling equipment could be used. Treatability studies would be required to define correct amendments, concentrations, mixing times, etc. Residuals generated would include relatively low volumes of decontamination fluids, PPE, etc. Costs would be low to moderate, which is relative to other ex situ treatment options.

This option is retained for further consideration.

9.5.6.7 Thermal Treatment Using Plasma Torch. This option would consist of vitrifying excavated contaminated soils and debris at high temperatures to produce a stable, glass-like inert wasteform. No reduction in radionuclide activity or toxic metal concentration would occur. Therefore, disposal in a low-level radioactive soil and debris landfill, or a RCRA Subtitle C or D landfill, of the wasteform would likely be required. This option alone would not reduce risks due to direct radiation exposure. Toxicity of the radionuclides and toxic metals would not be reduced. Availability of radionuclides and toxic metals, and exposure risks via soil ingestion and plant uptake, would be reduced. Mobility via leaching and infiltration to groundwater would be reduced. This technology would meet RCRA LDR treatment standards for toxic metal characteristic wastes cited in 40 CFR 268.40. This technology may offer little improvement in effectiveness over excavation and disposal in a secure landfill without treatment, but it is more administratively implementable.

Implementability of this option is considered low due to the technical complexity of the plasma torch process, including the requirement for an air pollution control system. Impacts to human health and the environment during operations could likely be reduced to allowable levels through administrative and engineering controls. Costs would be high.

Effectiveness is estimated as moderate. Costs are relatively high. This option is retained for further consideration.

9.5.6.8 Mercury Retort. This technology applies specifically to mercury-contaminated soils and sediments that fail, or are expected to fail, the RCRA TCLP test. The CFA-04 pond is the only site known at OU 4-13 where high-mercury subcategory RCRA characteristic soils may exist, however, none have been identified to date. Any excavated sediment failing the TCLP would have to be treated prior to disposal outside the AOC, under RCRA land disposal rules. Mercury retorting is specified as a best available technology for RCRA-hazardous nonwastewaters containing greater than 260-mg/kg total inorganic mercury in 40 CFR 268.40. No technology is specified for concentrations less than 260-mg/kg total inorganic mercury, and the TCLP allowable level of 0.20 mg/L is specified instead.

Mercury retorting consists of heating excavated contaminated soil to approximately 538°C (1,000°F) and volatilizing mercury as a vapor. The vapor is subsequently cooled and the liquid mercury recovered. Process equipment may include, but is not limited to, material handlers including a feed conveyer, heating units, heat exchangers, condensers, and air pollution control equipment including a baghouse and granular activated carbon absorbers. Recovered metallic mercury would be recycled.

This process has been used in previous INEEL removal actions to treat onsite CFA and Test Area North (TAN) soils contaminated with mercury in the 10- to 650-mg/kg range to residual concentrations that passed the TCLP. These operations generated large quantities of secondary wastes which included trash, discarded PPE, waste container liners, sanitary wastes, process condensate, scrubber water, and decontamination water. It is estimated that approximately 31 m^3 (40 yd^3) of secondary solid waste (sludge from the vapor recovery unit); 61 m^3 (80 yd^3) of contaminated PPE and trash, etc; and 83,000 L (22,000 gal) of secondary liquid wastes were generated while retorting approximately 344 m^3 (450 yd^3) of contaminated soil from CFA. The volume of secondary waste generated was approximately 50% of the volume of soil processed. These volumes could likely be reduced, based on lessons learned during these efforts.

Technical and administrative implementability of onsite retorting is considered moderate, since it has been implemented previously onsite at the INEEL. The availability of offsite retorting services, or equivalent treatment for high-mercury subcategory soils, is uncertain. No offsite capability was identified, based on responses to a recent LMITCO request for proposals (RFP) published in the Commerce Business Daily (CBD). Of the six responders to the request for offsite MLLW mercury retorting, none were both currently permitted and operating at full-scale. The retorted radioactive soil would likely have to be shipped to another facility after treatment. Implementability of off-INEEL mercury retorting is therefore uncertain. A facility in Tennessee appears the most likely vendor to offer MLLW mercury retorting at the scale required by 2000, and this facility was used as a representative off-INEEL treatment option.

Impacts to human health and the environment could likely be minimized to allowable levels through administrative and engineering controls. Costs are high. Both on- and off-INEEL mercury retorting are retained for further consideration for CFA-04 sediments and soils that fail the TCLP for mercury, and are contaminated at total Hg concentrations greater than 260 mg/kg.

9.5.7 Treatment In Situ

In situ treatment options are implemented without significant excavation of contaminated media. Construction requirements may include drilling wells, digging trenches, clearing and grubbing surfaces, removing existing structures, constructing aboveground process equipment, and other activities. Maximum remediation depths are assumed to be 3 m (10 ft) bgs, since no groundwater risks exist and all exposure pathways of concern would be addressed by remediating to this depth. In situ treatment options potentially applicable to OU 4-13 sites of concern are discussed below.

9.5.7.1 In Situ Chemical Stabilization Using Mechanical Mixing. This option would consist of using large-diameter augers, equipped with cutting blades and injection systems, to mix soils in situ at depths to at least 3 m (10 ft) bgs with chemical stabilization amendments to produce a stable, leaching-resistant wasteform. The drill unit can be covered with a shroud to control fugitive dust and collect off gas for processing, as required to reduce worker exposures. Augers are typically approximately 1.5 m (5 ft) in diameter and two are used simultaneously (EPA 1997b).

The effectiveness of this option in reducing risks to human health and the environment and in meeting RAOs is estimated as low for radionuclide-contaminated soils, and as moderate to high for toxic metal contaminated soils. This option alone would not eliminate risks to human health via direct radiation exposure. However, it may potentially reduce or eliminate risks due to homegrown produce ingestion and soil ingestion at OU 4-13 sites. Environmental risks would be reduced or eliminated by eliminating contaminant transport and/or exposure pathways. Toxicity of the COCs would not be reduced. Volume of contaminated materials would likely increase 30 to 50%, due to addition of amendments, which would raise the surface grade of the stabilized area several feet. Impacts to human health and the environment could likely be minimized to allowable levels through administrative and engineering controls.

Implementability of this option is moderate. Various methods of in situ chemical stabilization have been previously demonstrated at the INEEL, both for contaminated soils and landfilled waste (DOE 1995b). Costs are considered relatively moderate (\$20 to \$40 per yd^3 in 1997 dollars [EPA 1997b]).

Soil mixing is not cost-effective at sites where only surficial contamination occurs (e.g., CFA-10); or where contamination is relatively shallow (e.g., CFA-04), or where rock occurs at shallow depths within the melt zone (e.g., CFA-04). Soil mixing would be most technically implementable at CFA-08, where contamination extends to at least 3 m (10 ft) bgs. However, direct radiation exposure could still occur from the grouted product. Additionally, Cs-137 at CFA-08 will decay to unrestricted release levels (2.3 pCi/g) in 189 years, which results in overall low cost-effectiveness for soil mixing at this site. This option is screened from further consideration on the basis of low technical implementability and high cost.

9.5.7.2 In Situ Soil Washing. This process uses infiltration galleries or injection wells to advect extraction fluids through contaminated soils in situ. Downgradient wells recover the fluids for separation of the contaminants and reuse. Extraction fluids may include unamended water, acids or other oxidizers, surfactants, and others. This option would reduce or eliminate risks to human health and the environment from OU 4-13 sites by chemically removing contaminants for subsequent stabilization and disposal elsewhere. Toxicity of the COCs would not be reduced. Mobility of residual COCs would not be reduced without subsequent treatment and stabilization. Volume of contaminated materials might potentially be reduced, if effective.

As for ex situ soil washing, the effectiveness and technical implementability of this option is considered low. Soil washing, in combination with physical separation, has previously been tested at bench-scale for radionuclides including Sr-90 and Cs-137 on the INEEL TRA WWP sediments with poor results (INEL 1991). No soil washing treatability studies have been performed to date using toxic metal-contaminated INEEL soils. Treatability studies would be required to determine effectiveness for Pb, Cu, and Hg in OU 4-13 soils.

In situ soil washing is more technically complex than ex situ soil washing, due to the requirement for hydraulic control over the extractant fluid; and is likely less effective due to the difficulty of uniformly contacting the extractant fluid with contaminated media. Costs are considered moderate relative to other in situ treatment technologies. Impacts to human health and the environment would be minimal.

This option is screened from further consideration due to low technical implementability and low effectiveness.

9.5.7.3 In Situ Vitrification. In situ vitrification can potentially vitrify contaminated soils at depth to create a stable, glass-like mass. This technology is most commonly applied to soils contaminated to depths of at least 3 m (10 ft) bgs. The technology in situ vitrification (ISV) was developed by Batelle Pacific Northwest Laboratory, and is marketed exclusively by the Geosafe Corp., Richland, WA. A containment shroud is erected over the melting site and is maintained under negative pressure. The outlet of the shroud is connected to an off-gas treatment system consisting of a blower, and a combination of quenchers, scrubbers, mist eliminators, heaters, filters, and activated carbon adsorption specific to the site characteristics and contaminants.

Graphite electrodes are placed vertically in soil and large electrical currents applied to produce resistance heating. Power is supplied at the top of the soil initially. Flaked graphite and glass frit are placed on the soil to increase the electrical conductivity sufficiently to initiate melting. When melting begins, the electrodes are lowered 2.5 to 5 cm (1 to 2 in.) per hour, until the entire soil mass bounded by the electrodes is heated to 871 to 1,093 °C (1,600 to 2,000 °F) and melted. After cooling, the resulting wasteform is a leaching resistant glass-like form similar to obsidian. The process equipment is typically transported on three trailers, and is powered by utility power or a diesel generator. Typical power requirements are 800 to 1,000 kWh per ton of treated soil (EPA 1997b).

Volatile metals including mercury are volatilized during melting and captured by the off-gas treatment system, while less volatile metals including lead and arsenic tend to remain in the glass phase. Cesium may remain in the melt or volatilize, depending on the depth and time of the melting. The fate of silver is unknown. Volatile organics (VOCs) are vaporized or pyrolyzed by ISV. Vaporized VOCs that migrate to the surface are either burned in the hood covering the treatment area, or are treated in the off-gas treatment system (EPA 1994).

Site-specific considerations limiting technical implementability of ISV include: (1) void volume within a melt greater than 10%, (2) rubble greater than 20% by weight, and (3) combustible (organic) material greater than 5 to 10%. Soils with organic contents greater than 10% can reportedly be processed using lower melting rates and a larger scale off-gas treatment system. Recommended depth range of contamination for cost-effective implementation is 1.5 to 6 m (5 to 20 ft) bgs (EPA 1998).

The effectiveness of this option in reducing risks to human health and the environment and in meeting RAOs depends on the contaminants and exposure pathways of concern. This option alone would not significantly reduce risks to human health via direct radiation exposure, since most of the Cs-137 would remain in place. Immobilizing toxic metals in the glassy matrix would provide highly effective long-term containment, and would eliminate risks due to homegrown produce ingestion and soil ingestion for geologic time periods. Environmental risks would also be reduced or eliminated by eliminating exposure pathways. Mercury would primarily be volatilized, captured by the air treatment system, retorted to produce elemental mercury and shipped offsite for recycling, essentially eliminating risks due to mercury from the site. Residual amounts of mercury might remain in the glass at depth, but would be immobilized. Institutional controls may be required for long-term management of contaminants remaining at the site.

Toxicity of the radionuclides and toxic metals remaining in the melt would not be reduced; however, organics would be volatilized and captured, or destroyed by this method. Volume of contaminated soils could be reduced by as much as 20 to 45%, and the cooled melt surface could subside as much as 0.6 to 1.5 m (2 to 5 ft), relative to ground surface. Impacts to human health and the environment during implementation could likely be minimized to allowable levels through administrative and engineering controls.

Residuals produced by this process include metal vapor, organics, and solids captured on activated carbon, filters, or in scrubber solutions by the air pollution control system. Treatment media used to capture mercury could include a permanganate scrubber solution, or sulfur-impregnated carbon. Either type of media could be retorted onsite by the subcontractor to recover nonradioactive mercury for recycling, reducing the volumes of residual treatment media for disposal.

Implementability of this option is moderate to uncertain. Technical implementability is very site-specific. Melting rates of 5 to 6 m^3/hour (6 to 8 yd^3/hour) are reported (EPA 1997b). Costs are relatively high.

The ISV is not cost-effective at sites where only surficial contamination occurs (e.g., CFA-10); or where contamination is relatively shallow (e.g., CFA-04); or where rock occurs at shallow depths within the melt zone (e.g., CFA-04). The ISV would be most technically implementable at CFA-08, where contamination extends to at least 3 m (10 ft) bgs. However, Cs-137 at CFA-08 will decay to unrestricted release levels (2.3 pCi/g) in 189 years, which results in low cost-effectiveness for ISV at this site. This option is screened from further consideration on the basis of low technical implementability and high cost.

9.5.7.4 Phytoremediation.

Description. Phytoremediation is an innovative and emerging technology that utilizes surface vegetation to uptake toxic metals and radionuclides through roots and to degrade organic compounds in situ. Vegetation types may include grasses, vegetables, shrubs, trees, or other species. Metals incorporated in biomass may be recovered by harvesting the vegetation and incinerating the biomass. Incinerator residuals would require stabilization and disposal in a low-level radioactive waste, RCRA, or mixed-waste landfill.

Phytoremediation is most applicable for contaminants distributed within the rooting zone, typically 1 m (3 ft) maximum depth (EPA 1997). Parameters affecting application of this process include soil type and characteristics, contaminant type and chemical species, climate and others. Immobile precipitated contaminant species are not typically treatable by this method, without soil amendments. Soil amendments have included chelating agents like EDTA (Chaney et al. 1997), which can mobilize lead; and ammonium nitrate (DOE 1997b), which displaces exchangeable cations like Cs-137. Treatability studies are typically required to implement this technology successfully (EPA 1997).

Arthur (1982) observed radionuclide uptake in INEEL vegetation including Russian thistle, crested wheatgrass, and gray rabbitbrush growing on waste disposal sites, but did not quantitate uptake rates from soil. A number of plant species were evaluated for remediating low levels of Cs-137 and Sr-90 in soil at Brookhaven National Laboratory (BNL) (DOE 1997b). Hydroponic screening studies identified Reed canary grass (*Phalaris arundinacea*), Indian mustard (*Brassica juncea*), tepary bean (*Phaseolus acutifolius*), and cabbage (*Brassica oleracea*) as potentially hyperaccumulators of Cs-137. Subsequent studies in pots evaluated Cs-137 uptake from soil by these species. This study also evaluated soil amendments for releasing cesium sorbed to clay minerals, identified as a major impediment to phytoremediation of cesium. The most successful treatment consisted of amending soil with ammonium nitrate to promote release of cesium, allowing for subsequent uptake by cabbage. Cabbage grown in Cs-137-contaminated soils amended with 80-mole ammonium nitrate per kg soil showed bioaccumulation factors of approximately three, measured as activity of Cs-137 in dry shoot mass/Cs-137 in dry soil mass. This study indicated that reduction of initial Cs-137 soil activities of approximately 400 pCi/g to less than 100 pCi/g (75% activity reduction) using cabbage would take at least 15 years. The study also concluded that bioaccumulation ratios would decrease as activities decreased, making removal to lower activities unlikely in a reasonable time period.

Entry and Watrud (1998) determined that Alamo Switchgrass (*Panicum virginatum*) removed up to 44 and 36% of Sr-90 and Cs-137, respectively, from relatively shallow soil depths in pan (7 cm [3 in.] deep) and tube (30 cm [12 in.] deep) studies. Removal rates increased with increased soil radionuclide concentrations, and declined with successive plant harvests.

Argonne National Laboratory West (ANL-W) will begin a field demonstration in 1998. If successful, this technology would be applied to 15,000 m^3 (19,400 yd^3) of soils contaminated primarily with Cs-137 in the upper one foot of the soil column at ANL-W. After accumulating radionuclides, the vegetation would be harvested, sampled, and shipped to an incinerator on the INEEL for volume reduction. The resulting ash would then be sampled and sent to a permitted disposal facility.

A 1995 Argonne National Laboratory-East (ANL-E) study (DOE 1996) determined that *Phragmites australis*, a native plant determined to be tolerant of high toxic metal content soils, was able to uptake Pb but would not be able to reduce soil Pb initial concentrations ranging from 500 mg/kg to a 300 mg/kg release level within a reasonable time (20 years). Chemical species of lead present is likely very important in uptake rates; relatively soluble lead carbonate is reportedly accumulated by Festuca rubra (a grass), while elemental lead is likely not. Significant plant uptake of lead has not been demonstrated, but research utilizing soil amendments to increase uptake rates is reportedly underway (EPA 1997).

Effectiveness of this technology for OU 4-13 sites of concern is uncertain, because no treatability studies have been done to date for mercury, lead or Cs-137 in WAG 4 soils. However, results of previous studies are used in this section to provide a screening-level evaluation of effectiveness and technical implementability.

The maximum measured Cs-137 activity at CFA-08 is 180 pCi/g. Reduction to the PRG of 23.3 pCi/g would require an 87% activity reduction. Depth of contamination requiring remediation is 3 m (10 ft) bgs, which exceeds the 1 m (3 ft) upper limit of contaminant depths suggested by EPA (1997) by nearly a factor of three. Not all contamination is dispersed in soil; feeder lines and drain tiles still contain contaminated sludge, which would not be amenable to phytoremediation. Time to reach PRGs is unknown, because no treatability studies have been performed to date on WAG 4 soils and COCs. However, using the results of the DOE (1997b) study cited previously, over 15 years (minimum) would be required to attain the required Cs-137 activity reduction in the shallow treatment zone, using the best plant species and soil amendments identified in the study. Soil contamination at depths exceeding the rooting depth of the cabbage used would not be remediated, nor would sludges in feeder lines and drain tiles.

The maximum measured Hg concentration at CFA-04 is 439 mg/kg. Reduction to the ecological risk PRG of 0.74 mg/kg would require a 99.8% concentration reduction, which is likely unattainable. The depth of contamination is 0 to 0.9 m (0 to 3 ft) below the bottom of the pond (3 m [10 ft] bgs), which is potentially within the 1 m (3 ft) upper limit of contaminant depths suggested by EPA (1997). Genetically engineered plants have been developed specifically to uptake mercury from soil and groundwater (Flathman et al. 1998). However, mercury is not stored in biomass but is transpired and released to the atmosphere (i.e., volatilized). This "treatment" would not reduce toxicity (except through dilution), mobility or volume, and would merely dilute the mercury by dispersing it in air for eventual re-deposition to soil.

Technical implementability of this technology for WAG 4 sites is low to uncertain. The relatively short growing season of the INEELs sagebrush steppe environment constrains both species selection and biomass production. If nonarid climate vegetation species were used, which would likely be required to maximize biomass production, supplemental irrigation would likely be required, which could potentially flush mobile contaminants to depths greater than recoverable. This would also be a concern if supplemental soil amendments designed to increase uptake rates were used. Costs of this technology are estimated as low, relative to other in situ treatment technologies. Impacts to human health environment would be minimal. However, contaminants taken up by plants could be mobilized in the food chain during the treatment period, threatening environmental receptors.

Phytoremediation is screened from further consideration due to technical impracticability and lack of demonstrated effectiveness for INEEL soils and COCs.

9.5.8 Summary

Environmental monitoring process options retained include only soil monitoring. Institutional control actions include fences, deed restrictions, cap integrity monitoring and maintenance, and surface water diversion. The representative excavation technologies are standard equipment including backhoes and dozers. Robotics were rejected as not cost-effective. Field screening using gamma monitors and XRF would be used to the extent feasible to minimize the amount of uncontaminated soil removed.

Containment options retained include the SL-1-type and ET-type engineered barriers, the native soil cover, the RCRA-type cover and concrete.

Disposal options retained include the INEEL RWMC, the ICDF, an offsite MLLW landfill, an INEEL nonhazardous, nonradioactive disposal unit (e.g., the existing CFA landfill or the CFA-04 pond), and an offsite RCRA TSDF. The ICDF is retained pending a final decision on this project. The NTS was rejected from further consideration, since the INEEL is not currently an approved generator.

Ex situ treatment options for excavated radionuclide and metal-contaminated soils were evaluated based on their ability to reduce the overall toxicity, mobility and volume of contaminated soils at OU 4-13 sites. Technologies retained include segmented gate radioactive soil separation, chemical stabilization, plasma torch vitrification, and mercury retorting of mercury-contaminated soils.

Soil washing for removing Cs-137 was rejected on the basis of no demonstrated effectiveness for INEEL soils, as were flotation and attrition scrubbing. Soil washing for removing Pb, Cu and Hg was rejected, on the basis of low cost effectiveness. Screening was retained as a potential pretreatment step for other options.

No in situ treatment options were retained, due to low technical implementability, low effectiveness and high cost.

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